

Occurrence and Fate of Pharmaceuticals and Personal Care Products (PPCPs) in Biosolids

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ABSTRACT

Each year, large quantities of pharmaceuticals and personal care products (PPCPs) are used worldwide. Once conveyed to wastewater treatment plants, PPCPs can remain unchanged or undergo partial or complete transformation during wastewater treatment processes before discharge into the environment via effluent and biosolids for land application. Biosolids can be a major sink for some PPCPs. Previous investigations have indicated that land application of biosolids may be a potential important route through which PPCPs enter the environment. However, no information is available on exactly how closely the concentrations of PPCPs in the environmental media are related to the land application of PPCP-containing biosolids. This paper reviews currently available information on the occurrence of PPCPs in biosolids, methods of analysis, the potential fate of PPCPs in biosolids-applied soils, and composting as a potential means for removal of PPCPs from biosolids.

OVER THE LAST 40 years, efforts to understand the occurrence, fate, and environmental effects of anthropogenic chemicals have largely focused on industrial compounds and agricultural pesticides. The emphasis was appropriate due to the chemicals' large production, concentrated usage, widespread occurrence in the environment at high levels, persistence, and acutely toxic or carcinogenic effects. More recently, chemicals representing active ingredients in PPCPs have emerged as environmental contaminants with potentially widespread environmental effects.

A wide range of PPCPs has been detected in a variety of environmental samples at levels ranging from ng kg^{-1} up to g kg^{-1} (Halling-Sørensen et al., 1998; Daughton and Ternes, 1999; Kolpin et al., 2002). With the development of sophisticated and sensitive analytical instruments, more and more PPCPs can be detected at trace levels in the environment. Some of the detected PPCPs in the environment exhibit negative hormonal and toxic effects on various organisms at concentrations as low as $\mu\text{g kg}^{-1}$ (Daughton and Ternes, 1999). However, the environmental effects of many other PPCPs are not known. The concentrations, fate, and environmental effects of some PPCPs have been discussed in detail in a series of review articles (Halling-Sørensen et al., 1998; Daughton and Ternes, 1999; Jørgensen and Halling-

Sørensen, 2000; Daughton and Jones-Lepp, 2001; Snyder et al., 2003).

Different from industrial chemicals and agricultural pesticides, many PPCPs and their biological metabolites are bioactive and are introduced into the environment on a continual basis due to the worldwide frequent usage of PPCPs by humans or for animal husbandry (Daughton and Ternes, 1999). A variety of human use PPCPs are discharged into wastewater treatment plants (WWTPs) via excretion with urine and feces as parent compounds, conjugated compounds, or metabolites, and through washing or direct disposal. A variety of PPCPs have been detected at various concentrations in influents of WWTPs from different regions (Table 1). At WWTPs, PPCPs can remain unchanged or undergo transformation during the treatment processes before being discharged into the environment via effluent and biosolids. The pharmaceuticals administered for animal husbandry and the biological transformation products are excreted with urine and feces. The compounds can eventually enter into the environment due to runoff and leaching from feed lots and waste lagoons at confined animal feeding operations or from agricultural land irrigated or fertilized with animal waste (Halling-Sørensen et al., 1998).

Removal of PPCPs from wastewater via treatment in WWTPs can be substantial (30–90%) (Table 1). Whether this removal is due to solids partitioning or degradation is generally not known. Some PPCPs (e.g., nonylphenol) are not effectively degraded in WWTPs and accumulate in biosolids that can subsequently be disposed of on land. There is limited information on the concentrations of PPCPs in biosolids, resulting in a poor quantification of the contribution of biosolids land application to the occurrence of PPCPs in the environment. Due to the complex nature of biosolids, qualitative and quantitative analysis of PPCPs is challenging.

This paper reviews current information about the occurrence of PPCPs in biosolids, methods of analysis, the potential fate of PPCPs in biosolids-applied soils, and treatments that may reduce the levels of PPCPs in biosolids before introduction to the environment through land application.

PHARMACEUTICALS AND PERSONAL CARE PRODUCTS OF CONCERN

Each year, large quantities of pharmaceuticals are sold and consumed in the United States and worldwide for the diagnosis, treatment, alteration, or prevention of human diseases. From 1999 to 2002, pharmaceutical sales increased worldwide by about 25% to 424 billion

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Abbreviations: K_{ow} , octanol–water partition coefficient; NP, nonylphenol; PPCPs, pharmaceuticals and personal care products; WWTP, wastewater treatment plant.

Table 1. Concentrations of some pharmaceuticals and personal care products (PPCPs) in wastewater treatment plant (WWTP) influents and their percent decrease in effluents.

PPCP	Influent concentration	Decrease in effluent	Reference
	$\mu\text{g L}^{-1}$	%	
Acetylsalicylic acid	0.34 to 3.1	81 to 88	Ternes (1998), Heberer (2002)
Bezafibrate	1.2 to 5.3	27 to 83	Ternes (1998), Stumpf et al. (1999)
Caffeine	230	99.9	Heberer et al. (2002)
Carbamazepine	1.78 to 2.1	7 to 8	Ternes (1998), Heberer et al. (2002)
Ciprofloxacin	0.22 to 0.37	70 to 80	Alder et al. (2001)
Clofibric acid	0.46 to 1.2	0 to 51	Ternes (1998), Stumpf et al. (1999), Heberer et al. (2002)
Cyclophosphamide	0.007 to 0.143	0 to 94	Steger-Hartmann et al. (1997)
Diclofenac	0.035 to 3.02	-200 to 98	Ternes (1998), Stumpf et al. (1999), Heberer et al. (2001), Heberer et al. (2002)
Dimethylaminophenazone	1.1	38	Ternes (1998)
17 α -Ethinylestradiol	0.0002 to 0.013	-200 to 100	Ternes et al. (1999b), Baronti et al. (2000), Johnson et al. (2000)
Fenofibric acid	0.5 to 1.03	6 to 64	Ternes (1998), Stumpf et al. (1999)
Fragrances	0.3 to 154	80 to 100	Simonich et al. (2000)
Gemfibrozil	0.35 to 0.9	16 to 69	Ternes (1998), Stumpf et al. (1999)
Ibuprofen	0.3 to 4.1	90	Ternes (1998), Stumpf et al. (1999)
Indometacine	0.3 to 1.0	71 to 83	Ternes (1998), Stumpf et al. (1999)
Ketoprofen	0.6	48 to 69	Stumpf et al. (1999)
Metoprolol	6.5	83	Ternes (1998)
Naproxen	0.6 to 1.3	15 to 78	Ternes (1998), Stumpf et al. (1999)
Nonylphenol polyethoxylates	1.6 to 986	40 to 100	Keller et al. (2003)
Phenazone	0.3	33	Ternes (1998)
Propranolol	8.9	96	Ternes (1998)
Triclosan	0.5 to 1.3	34 to 92	Lindström et al. (2002)

Table 2. The most commonly used prescription and over-the-counter pharmaceuticals in the United States.

Active compound	CAS number	log K_{ow} [†]	Brand name	Use
Prescription drugs (top 10 prescribed in the United States in 2002) (RxList, 2004)				
Hydrocodone	125-29-1	0.98-2.45	Hydrocodone w/APAP	analgesic, antitussive, antipyretic
Acetaminophen	103-90-2	1.18-1.53		
Atorvastatin	134523-00-5	0.12-3.67	Lipitor	lipid-lowering agent
Atenolol	29122-68-7	0.23-1.37	Atenolol	beta1-selective (cardioselective) adrenoreceptor blocking agent
Levothyroxine	51-48-9	0.16-2.11	Synthroid	thyroid hormones
Estrone	53-16-7	3.22-3.38	Premarin	estrogens (female hormones)
Equilin	474-86-2	3.03-3.29		
17 α -Dihydroequilin	5965-19-5	6.21		
17 α -Estradiol	57-91-0	3.47-3.62		
Equilenin	517-09-9	2.95-3.42		
17 α -Dihydroequilenin	6639-99-2	3.12-3.55		
Azithromycin	83905-01-5	0.44-3.16	Zithromax	antibiotic
Furosemide	54-31-9	1.96-2.96	Furosemide	diuretic (treating hypertension, congestive heart failure, and edema)
Amoxicillin	26787-78-0	water soluble	Amoxicillin	gram-positive and gram-negative bactericide
Amlodipine	88150-42-9	0.26-3.38	Norvasc	treating high blood pressure and angina (diuretic)
Besylate	98-11-3	water soluble		
Hydrochlorothiazide	58-93-5	1.27-1.34	Hydrochloro-thiazide	diuretic and antihypertension
Common over-the-counter drugs (Arthritis Foundation, 2004; RxList, 2004)				
Acetaminophen	103-90-2	1.18-1.56	Anacin, Excedrin, Panadol, Tylenol	analgesic, anti-inflammatory
Ibuprofen	15687-27-1	0.82-3.40	Advil, Motrin IB, Nuprin	anti-inflammatory, analgesic, antipyretic
Aspirin	50-78-2	1.39-2.02	Anacin, Ascriptin, Bayer, Bufferin, Ecotrin, Excedrin tablets	analgesic, anti-inflammatory
Dextromethorphan	125-71-3	0.61-3.65	Benylin cough syrup	relieves cough
Diphenhydramine	58-73-1	0.27-3.34	Benadryl	antihistamine, cold and cough medicine
Loratadine	79794-75-5	4.56-4.77	Claritin	antihistamine
Omeprazole	73590-58-6	1.39-2.35	Prilosec	treating heartburn

[†] Octanol-water partition coefficient.

U.S. dollars (German Association of Research-Based Pharmaceutical Companies, 2004). In 2002, about 51% of the worldwide pharmaceutical sales were in the United States and Canada, 25% were in Europe, 12% were in Japan, 8% were in Africa, Asia, and Australia (not including Japan), and 4% were in Latin America (European Federation of Pharmaceutical Industries and Associations, 2003). Table 2 lists some of the most widely used pharmaceuticals in the United States. About 60% of the pharmaceuticals used in the United States are over-the-counter nonprescription drugs (Madhavan, 1994),

mostly for pain relief, cold and flu, allergy, etc. (Harris Interactive, 2002).

Most pharmaceuticals are designed to be nonbioaccumulative and eliminated from human or animal body shortly after administration. Once administered, metabolism of a pharmaceutical generally introduces hydrophilic functionalities onto the pharmaceutical molecule to facilitate excretion with urine and/or feces (Katzung, 2001).

In addition to pharmaceutical compounds, large quantities of personal care products, such as food supplements, fragrances, skin care and hair care products, insect repel-

Table 3. Common additives in some personal care products.

Additive compound	CAS number	log K_{ow} [†]	Characteristics
Fragrances			
Musk ketone	81-14-1	3.48	Distribution of the use of synthetic musks in personal care products: candles, air fresheners, and aroma therapy = 41%, perfumes, cosmetics, and toiletries = 25%, soaps, shampoos, and detergents = 34% (Fragranced Products Information Network, 2004).
Musk xylene	81-15-2	3.46	
Galaxolide (HHCB)	1222-05-5	4.60	
Tonalide (AHTN)	21145-77-7	4.84	
Phantolide (AHMI)	15323-35-0	4.53	
Traseolide (ATII)	68857-95-4	4.72	
Celestolide (ADBI)	13171-00-1	4.37	
Cashmeran (DPMI)	33704-61-9	4.84	
Flame retardants			
Tetrabromobisphenol A	79-94-7	0.39–5.34	Used as additive in flexible polyurethane foam, in textile coatings and coatings for furniture, and in plastics for electrical and electronic equipment, wire, and cable insulation and electrical connectors, automobiles, and construction and building materials (Bromine Science and Environmental Forum, 2004). The current estimated worldwide growth for flame retardants is 4% per year. Distribution of the 1.14 million Mg global consumption of flame retardants in 1998: Al-, Mg-, and N-based = 56%, Br-based = 23%, P-based = 15%, Cl-based = 6% (Clariant, 2004). Worldwide market demand for PBDEs in 2001 was 67 440 Mg, 83% of which was in the Americas (Hites, 2004).
Polybrominated diphenylether (commercial available PBDEs primarily consist of penta-, octa-, deca-PBDE)		log $K_{ow} \geq 5.74$, log $K_{ow} = 0.621(\text{Br}) + 4.12$ (Braekevelt et al., 2003)	
Polybrominated biphenyl		>4.0	
Pentabromochlorocyclohexane	87-84-3	4.01	
Hexabromocyclododecane	23774-70-1	4.98	
Pentabromotoluene	87-83-2	4.57	
Tetrabromophthalic anhydride	632-79-1	3.17	
Tris(2,3-dibromopropyl)phosphate	126-72-7	>4.0	
Disinfectants, antiseptics, and pesticides			
Triclosan (2,4,4'-trichloro-2'-hydroxy diphenyl ether)	3380-34-5	2.39–4.54	Bactericide added in detergents, dishwashing detergents, laundry soaps, deodorants, cosmetics, lotions, creams, toothpastes and mouthwashes, footwear, and plastic wear. It interferes with an enzyme crucial to the growth of bacteria (Bhargava and Leonard, 1996).
Biphenylol	90-43-7	2.67–2.98	Bactericide and virucide added in dishwashing detergents, soaps, general surface disinfectants in hospitals, nursing homes, veterinary hospitals, commercial laundries, barbershops, and food processing plants. It is used to sterilize hospital and veterinary equipment (National Library of Medicine Specialized Information Services, 2004).
Chlorophene	120-32-1	3.37–3.78	Bactericide and fungicide added in disinfectant solutions and soaps (National Library of Medicine Specialized Information Services, 2004).
DEET (<i>N,N</i> -diethyltoluamide)	134-62-3	2.44	Pesticide added in insect repellent (National Library of Medicine Specialized Information Services, 2004).
Butylparaben (alkyl- <i>p</i> -hydroxybenzoates)	94-26-8	1.49–3.26	Fungicide added in cosmetics, toiletries, and food (National Library of Medicine Specialized Information Services, 2004).
Surfactants			
Alkylphenol polyethoxylates (usually branched nonyl or octyl; ethoxylate units = 1–20)		>4.5	Nonionic surfactants added in detergents (National Library of Medicine Specialized Information Services, 2004).
Sodium dodecylbenzenesulfonate	25155-30-0	water soluble	Ionic surfactants added in detergents (National Library of Medicine Specialized Information Services, 2004).
Benzalkonium chloride	8001-54-5	water soluble	Ionic surfactants added in detergents, preservative and disinfectant in contact lens solutions (National Library of Medicine Specialized Information Services, 2004).

[†] Octanol–water partition coefficient.

lents, cleaning products, and flame retardants are produced and sold in large quantities worldwide each year (Table 3). Synthetic musks, for example, are a group of chemicals used as fragrances in many personal care products, such as soap, perfumes, detergents, shampoos, and other personal care products. Nitro musks and polycyclic musks are the most commonly used synthetic musks, and worldwide production in 1996 was 7600 Mg (Rimkus et al., 1999; Rimkus, 1999). One of the polycyclic musks, galaxolide (1,3,4,6,7,8-hexahydro-4,6,6,7,8,8-hexamethylcyclopenta[*g*]-2-benzopyran, HHCB), is included in the USEPA's list of high production volume (HPV) chemicals (those that are produced in or imported into the United States at ≥ 450 Mg yr⁻¹). Similarly, nonionic surfactants such as alkylphenol polyethoxylates (APnEOs) are widely used in many personal care products (Thiele et al., 1997). More than 200 000 Mg yr⁻¹ APnEOs, almost half of the world's annual production, were produced in the United States (United States International Trade Commission, 1995).

Many PPCPs are detected at wide concentration ranges in a variety of environmental samples (Halling-Sørensen et al., 1998; Daughton and Ternes, 1999; Kolpin et al., 2002; Guenther et al., 2002; Snyder et al., 2003; Richardson, 2003) and in animal tissues, human blood, and breast milk samples (Rimkus et al., 1994, 1999; Snyder et al., 2001; Adolfsson-Erici et al., 2002; Hites, 2004). Many of PPCPs exhibit negative hormonal and toxic effects on numerous organisms (Daughton and Ternes, 1999; Zerulla et al., 2002; Jjemba, 2002; Legler and Brouwer, 2003; Wilson et al., 2003). However, limited information is available on the effects of PPCPs on soil biota and plants. Jjemba (2002) prepared a list of PPCP compounds reported to be phytotoxic to certain plants. The reported phytotoxic concentrations in the growth medium ranged from 0.05 to 400 mg kg⁻¹ depending on plant species and the growth medium used. Fox et al. (2001, 2004) demonstrated that phytoestrogen signaling and symbiotic gene activation during plant–bacterial symbiosis can be disrupted by many endocrine-disrupting

chemicals, including nonylphenol (NP). Research by Gejlsbjerg et al. (2001) and Kollmann et al. (2003) failed to demonstrate negative effects of NP on bacterial denitrification, nitrification, aerobic respiration, and fungi activity in biosolids-amended soil.

Effluents of WWTPs are the most direct sources of PPCP contaminants to waterways (Halling-Sørensen et al., 1998; Calamari et al., 2003; Kolpin et al., 2004). However, PPCPs associated with land-applied biosolids can mobilize in soil and leach to ground water or enter surface water through runoff (Jjemba, 2002; Golet et al., 2003; Yang and Carlson, 2003; Pedersen et al., 2003).

FATE AND BEHAVIOR OF PHARMACEUTICALS AND PERSONAL CARE PRODUCTS DURING WASTEWATER TREATMENT PROCESSES

The average daily quantity of wastewater generated per capita in the United States is about 450 L (120 gallons), and contains approximately 240 mg L⁻¹ suspended solids (Hammer and Hammer, 2001). More than 80% of the suspended solids is organic matter. Typically, the larger the organic input to the WWTP, the greater is the amount of sewage sludge produced at the facility. Sewage sludge is digested through biological, chemical, and physical processes before it is dewatered to produce biosolids. A typical WWTP produces about 240 kg of dry biosolids per million liters of wastewater treated (Metcalf and Eddy, 1991).

Transformation of Pharmaceuticals and Personal Care Products during Wastewater Treatment Processes

The transformation of PPCPs in WWTPs varies with compound physicochemical properties and wastewater treatment conditions. During the wastewater treatment processes, the parent PPCPs, conjugates, and metabolites may be (i) completely transformed to CO₂, (ii) partially transformed producing metabolites, or (iii) unchanged (Jørgensen and Halling-Sørensen, 2000). Few studies have traced the fate of PPCPs during wastewater treatment processes. Most published work consists of laboratory incubation studies using biofilm reactors and batch reactors with activated sludge at room temperature (20–25°C). Studies conducted in laboratories may over- or underestimate transformation rates of PPCPs in WWTPs because temperature conditions in many WWTPs may be higher or lower than that simulated in the laboratory (Metcalf and Eddy, 1991). In addition, many operating conditions in WWTPs are difficult to simulate in the laboratory.

Destruction of PPCPs in wastewater has been looked at only recently and the results have been variable. Discussion on four examples is presented below for ibuprofen, 17 α -ethinylestradiol, diatrizoate, and cyclophosphamide together with the breakdown products.

The pain reliever ibuprofen (Table 2) was detected in the influents of several Switzerland WWTPs at concentrations of 1 to 3.3 μ g L⁻¹ (Buser et al., 1999). Its metabolites, hydroxyibuprofen (OH-Ibu) and carboxy-

ibuprofen (CA-Ibu) were detected in the influents at concentrations about 1.5 times higher than that of the parent compounds. However, much smaller concentrations (6–105 ng L⁻¹) of carboxyhydratropic acid (CA-HA), another metabolite of ibuprofen, were detected in the same influent samples. Aerobic laboratory results of the WWTP influent mixed with activated sludge indicated a rapid disappearance of these compounds from the liquid phase, to <1 to 3% of the original levels within 8 h. The authors suggested that no sorption of the test compounds by the sludge occurred during the incubation, but no experimental evidence was provided for this assertion. Laboratory incubation studies using batch reactors with activated sludge and synthetic wastewater (Zwiener et al., 2002) indicated that <10% of ibuprofen was degraded to OH-Ibu, CA-HA, and CA-Ibu within 50 h. A greater ibuprofen concentration (177 μ g L⁻¹) was used by Zwiener et al. (2002) than by Buser et al. (1999). The major metabolites of oxic and anoxic degradations of ibuprofen were OH-Ibu and CA-HA, respectively, but CA-Ibu occurred under both oxic and anoxic conditions. No further significant degradation of these three metabolites was observed (Zwiener et al., 2002). They suggested that remaining 90% of the ibuprofen that disappeared from the liquid phase of the reactors may have been sorbed by the sludge, partially degraded to other metabolites, or completely degraded. It was observed that the efficiency of elimination of ibuprofen from biofilm reactors decreased with increasing initial ibuprofen concentration.

The active ingredient of contraceptive pills, 17 α -ethinylestradiol, is eliminated from the human body as conjugates with glucuronic acid and sulfate (Ranny, 1977). Once discharged into WWTPs, these conjugates cleave fairly rapidly in contact with the microorganisms in activated sludge to release 17 α -ethinylestradiol, which is stable under aerobic conditions of an activated sludge process (Ternes et al., 1999a). This conceptualization may explain why the concentration of 17 α -ethinylestradiol in the effluent of a German WWTP was detected at a level almost twice as great as in the influent (Ternes et al., 1999b). Compared with other steroidal hormones, 17 α -ethinylestradiol degradation in an undiluted mixed sewage liquor was only 20% over 24 h, whereas 75% of 17 β -estradiol, a natural hormone, mineralized over 24 h (Layton et al., 2000).

Diatrizoate incubated with fresh activated sludge for 2 wk was not degraded, whereas 85% of iopromide was transformed into two metabolites within 54 h (Kalsch, 1999). Both diatrizoate and iopromide are widely used X-ray contrast media for detailed images of soft tissues in X-ray radiography. Cyclophosphamide, one of the most frequently used agents in cancer chemotherapy, exhibited poor degradability during 30-d batch incubation with activated sludge (Steger-Hartmann et al., 1997).

Large volumes of wastewater (up to hundreds million liters) in a typical WWTP must be treated every day and the average wastewater retention time (hydraulic retention time) in a WWTP varies from <1 h to a few days. Explaining the wastewater treatment process further is beyond the scope of the paper and standard texts

in the field are available for review (e.g., Metcalf and Eddy, 1991; Hammer and Hammer, 2001). Hydraulic retention time frames are generally shorter than the degradation half-lives of many PPCPs that enter WWTPs (Halling-Sørensen et al., 1998), resulting in discharge of some relatively soluble PPCPs in effluent before degradation can occur.

Most studies on the fate of contaminants during WWTP processes have focused on the concentrations of target compounds in effluent discharged to the environment. A WWTP is considered to have efficient removal if there is a significant concentration reduction for the compound of interest in effluent compared with concentrations in influent (Simonich et al., 2002). This concentration reduction in effluent may be due to sorption of the target compounds to the solid phase, rather than degradation in the solution phase (Keller et al., 2003; Golet et al., 2003).

Occurrence of Pharmaceuticals and Personal Care Products in Biosolids

Compared with the hydraulic retention time of the liquid phase, the residence time of biological solids (mean cell residence time) in a WWTP system is much greater, ranging from a few days up to 30 d, due to repeated recycling of biological growths and extracted waste organic matter from one treatment reactor to another (Hammer and Hammer, 2001). However, even this residence time is shorter than the half-lives of some PPCPs (Halling-Sørensen et al., 1998). More importantly, during the wastewater treatment processes, PPCP molecules can move into microsites within the solid phase matrix (an "aging" process), and the biodegradability of organic compounds can be reduced significantly (Hatzinger and Alexander, 1995; Kelsey et al., 1997; Nam et al., 1998; Alexander, 2000). Sequestration into microsites of the solid phase can slow or stop microbial degradation if the molecules are inaccessible to microorganisms or extracellular enzymes. Diffusion of sequestered molecules out of the microsites in the solid phase can be extremely slow (Hatzinger and Alexander, 1997).

During wastewater treatment, PPCP and metabolite compounds partition into the solid phase depending on compound hydrophobicity, which is related to the compound octanol-water partition coefficient (K_{ow}). The greater the K_{ow} of a compound, the more hydrophobic it is. Dobbs et al. (1989) found that the sorption of certain chlorinated organic compounds on primary, mixed-liquor, and digested solids from municipal wastewater treatment plants correlated positively with their $\log K_{ow}$, ranging from 1.26 to 5.48 (Fig. 1).

The organic matter contents of sludge solid phases vary from 40 to 85% on dry-weight basis, resulting in large surface areas ($0.82\text{--}1.66\text{ m}^2\text{ g}^{-1}$) (Strachan et al., 1983; Wang et al., 1993). Wang et al. (1993) proposed a two-stage mechanism for sorption of toxic organic compounds on the sludge solid phase: initial adsorption on the surface of the sludge, followed by partitioning into the interior of the sludge biomass. The $\log K_{ow}$

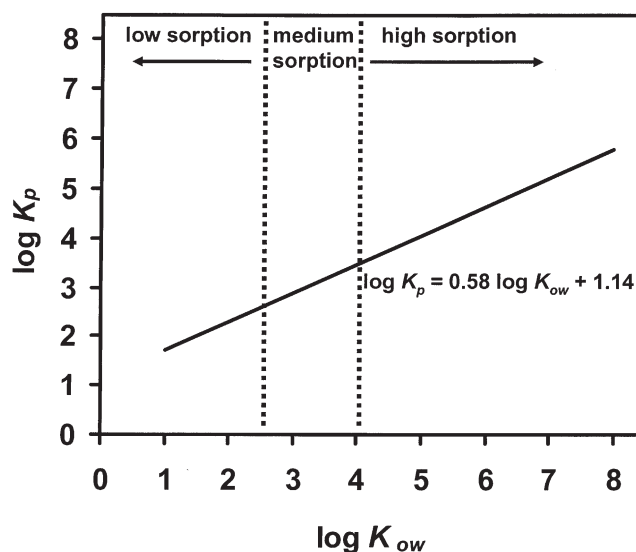


Fig. 1. Experimental correlation between octanol-water partition coefficient (K_{ow}) of some chlorinated organic compounds and their partition coefficients (K_p) for sewage sludge (based on weight of organic matter in the sludge solid phase). Vertical dotted lines indicate the boundary for sorption tendency of organic compounds on the sludge solid phase (data adopted from Dobbs et al., 1989; Rogers, 1996, with permission).

values for the compounds used by Wang et al. (1993) were similar to that used by Dobbs et al. (1989) (Fig. 1).

Although little research has been found on the sorption of PPCPs on the sludge solid phase during wastewater treatment processes, the relationship between PPCP sorption on the sludge solid phase and compound K_{ow} values can be expected to be similar as that indicated in Fig. 1. As shown in Tables 2 and 3, many PPCPs have K_{ow} values in the range of medium to high tendency for sorption on the sludge solid phase (Fig. 1). The sorption mechanisms proposed by Wang et al. (1993) for toxic organic compounds in the sludge solid phase should also apply to the PPCPs with similar K_{ow} values. Previous laboratory and full-scale wastewater treatment studies suggest that sorption to sewage sludge is the main removal process for some PPCPs from the wastewater stream (Kümmerer et al., 2000; Golet et al., 2003; Keller et al., 2003).

There is limited information on levels of PPCPs in biosolids. Fragrances have been detected at levels ranging from 1.5 to $147\text{ }\mu\text{g kg}^{-1}$ (dry mass) in biosolids from the United States, Switzerland, and the Netherlands (Berset et al., 2000; Difrancesco et al., 2004). Nonylphenol polyethoxylates (NPnEOs) and nonylphenol (NP), a metabolite of NPnEOs, have been detected at concentrations as high as 981 mg kg^{-1} (dry mass) and 1380 mg kg^{-1} (dry mass), respectively, in biosolids from many U.S. states (La Guardia et al., 2001; Keller et al., 2003; Xia and Pillar, 2003). Brominated diphenylethers, commonly used fire retardants, have been detected at 32 to $4890\text{ }\mu\text{g kg}^{-1}$ (dry mass) in biosolids from several WWTPs in the United States and the Netherlands (Hale et al., 2001; De Boer et al., 2003). Golet et al. (2002) detected 1.4 to 2.4 mg kg^{-1} (dry mass) of fluoroquino-

lone antibacterial agents in biosolids samples from Switzerland.

Approximately 6.9 million dry Mg of biosolids were generated in the United States in 1998, and production is estimated to reach 8.2 million dry Mg by 2010 (USEPA, 1999). Methods for disposal of biosolids include landfill, incineration, and land application. By 2010, 48% of biosolids are predicted to be land-applied (USEPA, 1999). Land application of biosolids, containing contaminants sorbed to biosolids, may concentrate in soil over time, providing a reservoir of pollutants that could eventually enter waterways through leaching and runoff. Hence, there is a need to evaluate the presence and environ-

mental fate of PPCPs in biosolids before the occurrence of detrimental effects to the environment.

ANALYTICAL METHODS FOR PHARMACEUTICALS AND PERSONAL CARE PRODUCTS IN ENVIRONMENTAL SAMPLES

Numerous PPCPs have been detected at concentrations as low as in the ng L^{-1} range in a variety of environmental samples including surface water, ground water, drinking water, and sediments samples. Several review papers on analytical methods for PPCPs in water and

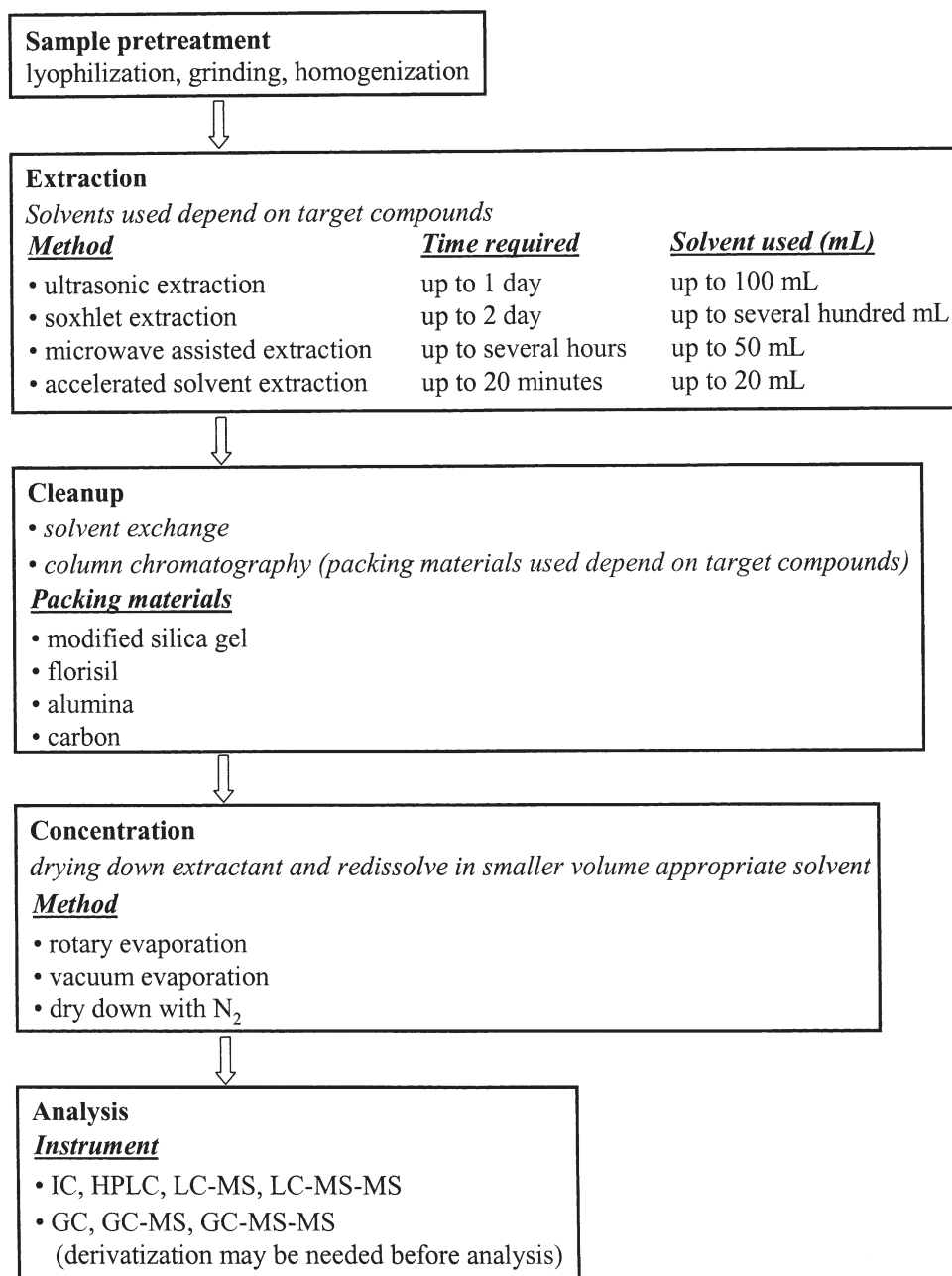


Fig. 2. General procedures for pharmaceuticals and personal care products (PPCPs) analysis in sediment samples. A summary of methods is cited in Table 4. GC, gas chromatography; HPLC, high performance liquid chromatography; IC, ion chromatography; LC, liquid chromatography; MS, mass spectrometry.

sediment samples have been published (Simonich et al., 2000; Golet et al., 2001; Öllers et al., 2001; Petrović et al., 2001; Sacher et al., 2001; Ternes, 2001; Hyötyläinen and Hartonen, 2002; Patterson et al., 2002), but there is limited information on methods for PPCPs analysis in biosolids (La Guardia et al., 2001; Golet et al., 2002; Patterson et al., 2002). The published methods on water and sediment samples can be a good foundation for biosolids method development.

Figures 2 and 3 illustrate the general procedures for extraction, cleanup, concentration, and analysis for sediment and water samples. Among all the extraction methods for sediments, accelerated solvent extraction (ASE) is the most robust. It is operated under high pressure and at a temperature that can be programmed to the optimum temperature for extraction, resulting in efficient and rapid extractions using a small volume of solvent (Richter et al., 1996). In addition, a gradient of

multiple solvents can be used for ASE to extract compounds with varying hydrophobicities. A combination of adsorbents is packed in an open column before an extractant is passed through for cleanup. The most commonly used packing materials are modified silica, Florisil, alumina, and different types of carbon. Once interferences from the matrix are removed through cleanup steps, the sample is normally dried down and redissolved in a smaller volume of solvent to increase the concentration of the target compounds. Compounds in the concentrated samples are then identified and quantified using a variety of instruments. Liquid chromatography–mass spectrometry (LC–MS, LC–MS–MS) and gas chromatography–mass spectrometry (GC–MS, GC–MS–MS) are the most powerful tools because of their identification capabilities and high sensitivities. More specific procedure requirements for selected PPCPs are listed in Table 4.

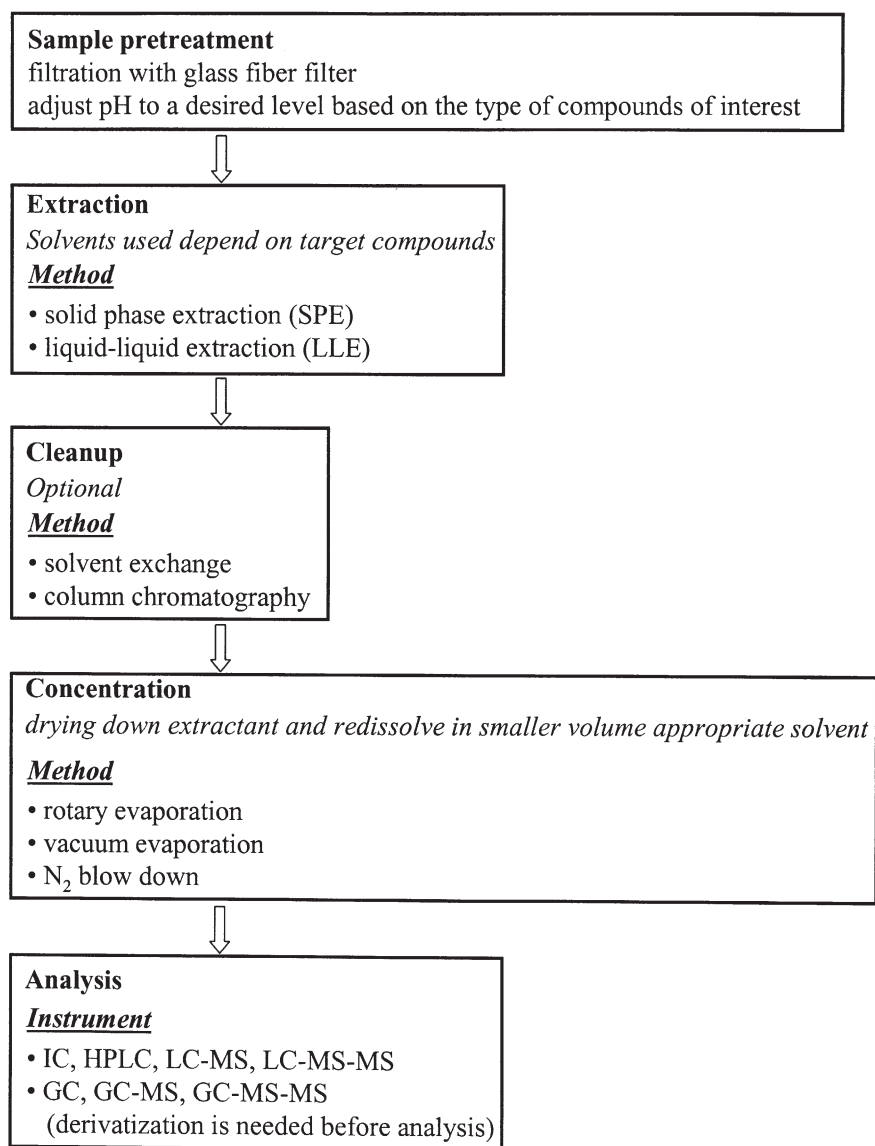


Fig. 3. General procedures for pharmaceuticals and personal care products (PPCPs) analysis in water samples. A summary of methods is cited in Table 4. GC, gas chromatography; HPLC, high performance liquid chromatography; IC, ion chromatography; LC, liquid chromatography; MS, mass spectrometry.

Table 4. Methods for selected pharmaceuticals and personal care products (PPCPs) analysis in environmental samples. The general procedures are outlined in Fig. 2 and 3.[†]

Analyte	Matrix	Pretreatment	Extraction	Cleanup	Analysis	Reference
Musk ketone	liquid sewage	none	liquid-liquid with hexane	gel permeation chromatography, elution with hexane-ethyl acetate (1:1), silica SPE eluted with CH ₂ Cl ₂	GC-MS-MS, first MS, <i>m/z</i> = 279, second MS, <i>m/z</i> = 191, 262	Berset et al. (2000)
Musk xylene	wastewater	none	C ₁₈ SPE eluted with CH ₂ Cl ₂	none	GC-MS-MS, first MS, <i>m/z</i> = 282, second MS, <i>m/z</i> = 265, 280	Simonich et al. (2000)
HHCB	solid sludge	homogenize with hydromatrix	ASE with CH ₂ Cl ₂ at 60°C, 1.4 × 10 ⁵ Pa	activated silica solvent exchanged to hexane	GC-MS, <i>m/z</i> = 243, 213	
HHCB, AHTN	freeze-dried, sieved through 2 mm	freeze-dried, sieved through 2 mm	ASE with CH ₂ Cl ₂ at 100°C, 6.9 × 10 ⁶ Pa	size exclusion, solvent exchanged to hexane, activated silica eluted sequentially with hexane, hexane-CH ₂ Cl ₂ (6:4), acetone, solvent exchanged to toluene	GC-MS, <i>m/z</i> = 258, 243 same as above	
NP1EO	biosolids	freeze-dried, sieved through 2 mm	ASE with CH ₂ Cl ₂ at 100°C, 6.9 × 10 ⁶ Pa	activated silica solvent exchanged to hexane	GC-MS, <i>m/z</i> = 179	La Guardia et al. (2001)
NP2EOs	wastewater	filter and acidify pH to <2	C ₁₈ SPE eluted with methanol	none	GC-MS, <i>m/z</i> = 223	Keller et al. (2003)
NP, NPnEOs	sediment	centrifugation	shake with acetone and then with acetone-hexane	washed with NaCl-NaH ₂ PO ₄ buffer, sulfur removed with mixture of 2-propanol and TBA sulfite reagent	HPCL-FLD-UV	Sellström et al. (1998), Alchini et al. (1999), Covaci et al. (2003)
PBDEs	sediment	centrifugation	shake with acetone and then with acetone-hexane	silica gel column eluted with hexane-acetone (65:35) derivatized with MSTFA-TMSI-DTE (1000:22:2) for 0.5 h at 60°C before GC-MS-MS analysis	GC-MS-MS, first MS, <i>m/z</i> = 342, second MS, <i>m/z</i> = 257, 244	Ternes et al. (1999a, 1999b)
Estrone	wastewater	filter through glass fiber, adjust pH to <3	RP-C ₁₈ SPE eluted with acetone	none	GC-MS-MS, first MS, <i>m/z</i> = 416, second MS, <i>m/z</i> = 326, 285	Zhu et al. (2001)
17β-Estradiol	water	adjust pH to 2.5	SPE with C ₁₈ Sep-Pak (Waters, Milford, MA) eluted with 10 mM oxalic acid in methanol and SPE Oasis HLB (Waters) cartridges, eluted with 1% TFA in methanol	none	GC-MS-MS, first MS, <i>m/z</i> = 425, second MS, <i>m/z</i> = 231, 193	
Oxytetracycline	water	adjust pH to 2.5	SPE with C ₁₈ Sep-Pak (Waters, Milford, MA) eluted with 10 mM oxalic acid in methanol and SPE Oasis HLB (Waters) cartridges, eluted with 1% TFA in methanol	none	LC-MS-MS with ESI C ₁₈ column, mobile phase of H ₂ O-5% formic acid-acetonitrile-methanol (23:40:25:12), first MS, <i>m/z</i> = 445, second MS, <i>m/z</i> = 410	
Chlortetracycline	water	adjust pH to 2.5	SPE with C ₁₈ Sep-Pak (Waters, Milford, MA) eluted with 10 mM oxalic acid in methanol and SPE Oasis HLB (Waters) cartridges, eluted with 1% TFA in methanol	none	first MS, <i>m/z</i> = 461, second MS, <i>m/z</i> = 426	
Benzalkonium chlorides	sediment	freeze-dry	ASE with acetonitrile-water (6:4) at 120°C, 3.4 × 10 ⁶ Pa	cleanup and concentration using an automated SPE system with PLRP-s cartridges (Polymer Laboratories, Amherst, MA) eluted with acetonitrile-water (6:4)	first MS, <i>m/z</i> = 479, second MS, <i>m/z</i> = 444	Ferrer and Furlong (2002)
Clofibric acid	wastewater	4 L of water was basified to pH = 12-14, extracted with CH ₂ Cl ₂	aqueous phase was acidified to pH = 1, extracted with CH ₂ Cl ₂ , dried down to 2.4 mL	none	LC-MS-MS with ESI RP-C ₁₈ column with mobile phase of acetonitrile-10 mM ammonium formate (from 50:50 to 100:0 at 15 min), <i>m/z</i> = 304, 332, 360	Patterson et al. (2002)
Fluoroquinolone	wastewater	filtered through cellulose nitrate filter (0.45 μm), acidified to pH = 3	SPE with mixed-phase cation exchange disk cartridge eluted with 5% ammonia solution in 15% methanol	none	GC-MS, <i>m/z</i> = 228, 230, 128, 130, 169, 171	Golet et al. (2001)
Acidic drugs (salicylic acid, ibuprofen, keto-profen)	water	filtered with glass fiber filter and acidified to pH = 2	Oasis HLB cartridge eluted sequentially with acetone, methanol, and acetone, dried down and redissolved in methanol	none	LC-MS-MS with ESI RP-C ₁₈ column with mobile phase of 0.1% TFA aqueous solution-acetonitrile (88:12 to 85:15 in 15 min), first MS, <i>m/z</i> = 332, second MS, <i>m/z</i> = 314, 288, 245, 231	Farré et al. (2001)
Neutral pharmaceuticals (see reference listed in the right column)	water	filtered through glass fiber, neutralized to pH = 7-7.5	C ₁₈ SPE eluted with methanol	none	LC-MS with ESI ions monitored can be found in the reference listed on the right column	Ternes et al. (2001)
					LC-MS-MS with ESI LiChrospher RP-18 column (Agilent Technologies, Palo Alto, CA) with mobile phase of 20 mM ammonia acetate in water-acetonitrile, ions monitored can be found in the reference listed on the right column	

[†] AHTN, tonalide; ASE, accelerated solvent extraction; DTE, dithioerythritol; ESI, electron spray ionization; FLD, fluorescence detector; GC, gas chromatography; HHCB, galaxolide; HPCL, high performance liquid chromatography; LC, liquid chromatography; MS, mass spectrometry; MSTFA, N-methyl-N-(trimethylsilyl)-trifluoroacetamide; NP, nonylphenol; NP, NPnEOs, nonylphenol polyethoxylates; PBDEs, polybrominated diphenylethers; RP, reverse phase; SPE, solid-phase extraction; TBA, tetrabutylammonium; TFA, trifluoroacetic acid; TMSDM, trimethylsilyldiazomethane; TMSI, trimethylsilylimidazole; UV, ultraviolet.

FATE OF PHARMACEUTICALS AND PERSONAL CARE PRODUCTS IN BIOSOLIDS AND BIOSOLIDS-AMENDED SOILS

Fate of Pharmaceuticals and Personal Care Products in Biosolids-Amended Soils

Land application of PPCP-containing biosolids can provide important routes through which the residual PPCPs enter the environment. The potential for surface runoff and leaching can affect PPCP transport into surface and ground water. The decay curves shown in Fig. 4 were used by Beck et al. (1996) to describe five possible fates for organic chemicals, such as chlorinated benzenes, polychlorinated dibenzo-*p*-dioxins, polychlorinated benzenes, and polyaromatic hydrocarbons in biosolids-amended soils. The same fates may also apply to biosolids-associated PPCPs, which exhibit a wide range of hydrophobicity (Tables 2 and 3). To date, there is limited information on the fate and transport of PPCPs in biosolids-amended soil.

Hesselsøe et al. (2001) demonstrated that the aggregate size of biosolids affects oxygen availability and, therefore, the aerobic transformation of organic contaminants such as nonylphenol (NP) in biosolids-amended soils. Degradation of NP was complete within 38 d in homogenous mixtures of soil and biosolids aggregates, but was retarded to more than 120 d in nonhomogeneous mixtures. A laboratory study by Xia and Jeong (2004) demonstrated a 55% reduction of NP in surface-applied biosolids within 30 d of light exposure. The data suggest that sensitized photolysis reactions may play important roles in degrading NP in surface-applied biosolids. Söderström et al. (2004) observed that decabromodiphenyl ether, a flame retardant, was debrominated photolytically in a soil matrix to form lower brominated bromodiphenyl ether compounds that were more resistant to photolytic degradation and more bioaccumulative. The half-life of photodegradation of decabromodiphenyl ether in soil was estimated to be between 150 to 200 h.

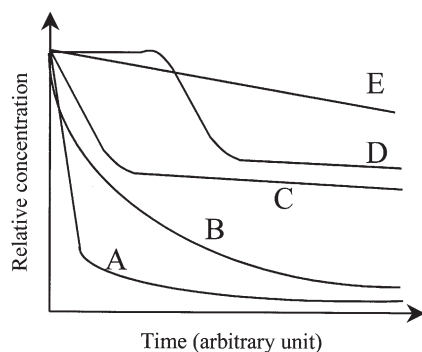


Fig. 4. Conceptual trend for the persistence of organic chemicals in biosolids-amended soils. Curve A is for compounds that are volatile, water soluble, or easily degraded; Curves B and C are for compounds with intermediate sorption potential on biosolids; Curve D is for compounds whose initial degradation is rate-limited because the microbial population has not acclimated to the compounds; and Curve E is for compounds that are nonvolatile, relatively water insoluble, and recalcitrant (Beck et al., 1996, with permission).

Difrancesco et al. (2004) observed rapid dissipation of 22 fragrance compounds 3 mo after being spiked in biosolids-amended soils. Complete dissipation for most of the tested compounds, except musk ketone and AHTN, was observed within the 1-yr die-away experiment, using soils with a wide range of textures and organic matter contents. A laboratory batch study conducted by Kreuzig et al. (2003) suggested that the abiotic transformation of diclofenac in a clayey silty soil and a silty sandy soil was dominated by the formation of non-extractable residues and that photo-induced degradation less relevant. Both soils were not amended with biosolids. Matscheko et al. (2002) investigated the levels of polybrominated diphenylethers (PBDEs) in lands that received biosolids application. Elevated levels of PBDEs were detected in all the biosolids-amended soils, compared with background levels. Twenty years after application of biosolids at 25 Mg ha⁻¹ yr⁻¹ for four consecutive years, the concentration of total PBDEs was 840 µg kg⁻¹, almost 8000 times the background level in the area. The concentrations of the tested compounds in the biosolids applied were not provided by the authors. Significant accumulation of PBDEs in earthworms of the sites was also observed. This study suggests that PBDEs are persistent in the soil environment, perhaps mainly due to the compound's high hydrophobicity (Table 3).

Insignificant mineralization (approximately 0.49–0.58%) of sarafloxacin, a fluoroquinolone antibiotic, was observed in a loam soil, a silt loam soil, and a sandy loam soil after 80 d incubation under aerobic conditions (Marengo et al., 1997). Biosolids were not applied to those soils. Marengo et al. (1997) concluded that the minimal mineralization may have been due to strong binding of sarafloxacin to the soil and the consequent nonavailability to soil microorganisms. Colucci and Topp (2001) observed rapid microbial-mediated dissipation of 17α-ethinylestradiol under aerobic condition in soils varying widely in texture and properties. The 17α-ethinylestradiol dissipated to below the detection limit (50 µg kg⁻¹) within 22 d of incubation. The 17α-ethinylestradiol (log *K*_{ow} = 3.8) is much more hydrophobic than sarafloxacin (log *K*_{ow} = 0.49).

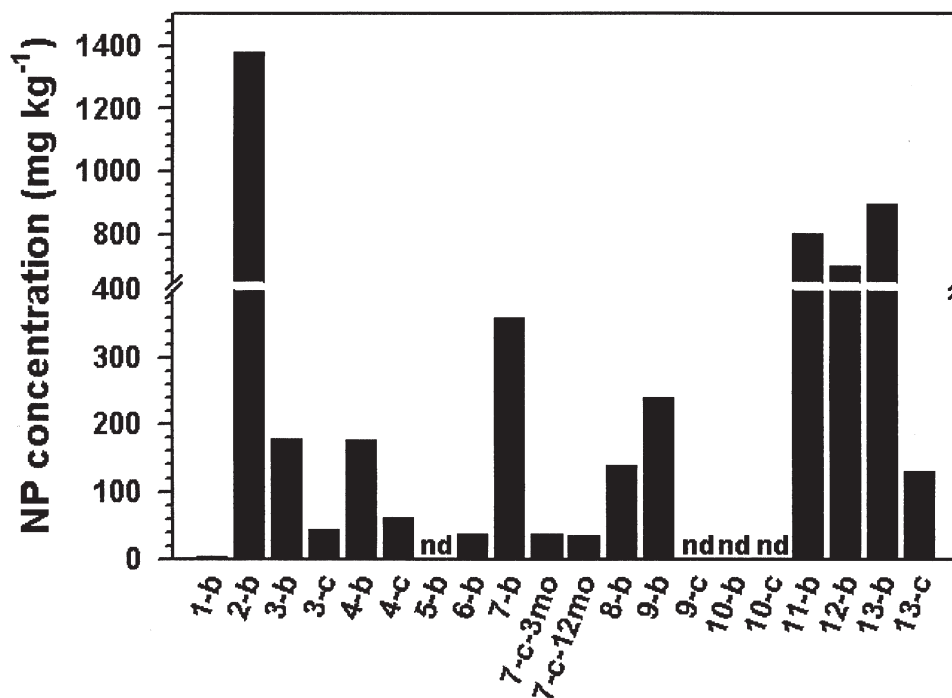
Tolls (2001) determined the sorption of a variety of veterinary pharmaceuticals with log *K*_{ow} ranging from -2.34 to 3.5 in soils without biosolids amendment. The sorption coefficients to soils varied from 0.2 to 6000 L kg⁻¹, demonstrating that these chemicals display a wide range of mobility. The author suggested that mechanisms in addition to hydrophobic partitioning played a significant role in sorption of the tested compounds in soils. For the highly hydrophilic compounds, ion exchange, ion bridging at clay surfaces, surface complexation, and hydrogen bonding appeared to be involved in their sorption in soils. Inevitably, those reactions are largely affected by soil pH, soil mineral composition, and soil solution chemistry. Tolls (2001) proposed that instead of merely considering the contribution of hydrophobic partitioning to sorption in soils, a conceptually more complete representation of sorption of pharmaceutical compounds should be adopted.

There have been few investigations on the mobility and transport of PPCPs in soils, especially biosolids-amended soils. Rabølle and Spliid (2000) reported that the weakly adsorbing olaquinox (an antibiotic, $\log K_{ow} = 0.11$) completely leached through soil columns, whereas the stronger-adsorbing antibiotic tylosin ($\log K_{ow} = 3.14$) was retained in different depths depending on the soil properties. Boxall et al. (2002) observed that sulfachloropyridazine, an ionic antibiotic with low sorption coefficients ($0.9\text{--}1.8 \text{ L kg}^{-1}$), was rapidly transported to surface waters after application to land. Thiele-Bruhn (2003) suggested that fast leaching through soils by macropore or preferential transport facilitated by dissolved soil colloids was the major transport process for strongly sorbed PPCPs. Surface runoff from lands may also contribute to widespread detection of PPCPs in the water environment. A variety of PPCPs were

identified by Pedersen et al. (2003) in surface runoff from agricultural fields irrigated with disinfected tertiary recycled water or wastewater effluent-dominated stream water.

Removal of Pharmaceuticals and Personal Care Products in Biosolids through Composting

The safest approach to avoid potential detrimental effects of biosolids-associated PPCPs to the environment is to ensure that the compounds are adequately degraded before biosolids land application. Composting has been used as an effective means to degrade xenobiotic organic contaminants such as pesticides, polycyclic aromatic hydrocarbons (PAHs), polychlorinated biphenyls (PCBs), trinitrotoluene (TNT), and perchlorate (Williams and Keehan, 1993; Rao et al., 1995; Wallace



Characteristics of the WWTPs investigated				
WWTP	Location	Population	Community	Treatment
1	GA	33,000	Mostly residential	Aerobic
2	SC	125,000	Mostly industry	Anaerobic
3	SC	30,000	Mostly residential	Aerobic
4	GA	60,000	Mostly residential	Aerobic
5	SC	50,000	Mostly residential	Aerobic
6	GA	25,000	Mostly residential	Aerobic
7	GA	40,000	Mostly industry	Anaerobic
8	GA	70,000	Urban residential area	Aerobic
9	GA	30,000	Mostly residential	Anaerobic
10	GA	29,000	Residential and industry	Aerobic
11	GA	600,000	Urban area and industry	Anaerobic
12	GA	175,000	Residential and industry	Anaerobic
13	KS	150,000	Urban area and industry	Anaerobic

Fig. 5. Concentrations of nonylphenol (NP) in biosolids and composts from 13 wastewater treatment plants (WWTPs) in Georgia, South Carolina, and Kansas. The numbers indicate the WWTPs listed in the table below the graph. b, Biosolids; c, compost; c-3mo, biosolids composted for 3 months; c-12mo, biosolids composted for 12 months; nd, not detectable (Xia and Pillar, 2003).

et al., 1998; Büyüksönmez et al., 1999; Weed et al., 1999). Composting may accelerate the degradation of organic contaminants due to their exposure to high microbial diversity and activity (especially thermophilic organisms), abundant substrates, high temperature, changing pH, and successive shifts in aerobic and anaerobic conditions in microenvironments within a composting system (Büyüksönmez et al., 1999; Barker and Bryson, 2002).

Results from a recent investigation of 13 WWTPs in three U.S. states showed significantly lower concentrations of NP (65–100% less) in composted biosolids than in fresh biosolids (Fig. 5). A pilot laboratory-controlled composting study (Fig. 6) provided further evidence that NP can be effectively degraded during composting. Biosolids initially containing approximately 450 mg kg^{-1} NP was mixed with wood shavings at dry weight ratios of 43:57, 65:35, and 84:16. The mixtures were then incubated aerobically at 25, 45, and 65°C with the moisture content maintained at 65% for up to 70 d. A temperature between 45 and 65°C is ideal for thermophilic microorganisms, while 25°C is an ideal temperature for mesophilic microorganisms (Büyüksönmez et al., 1999). In all treatments, a rapid degradation of NP occurred within 15 d of incubation, with a maximum removal rate of 80%. Temperature significantly affected NP degradation at early stage of incubation. For compost with a biosolids to wood shavings ratio of 43:57, 6 d of incubation at 65°C resulted in a 76% NP reduction, whereas 41% of NP was degraded after an 8-d incubation for treatments with biosolids to wood shavings ratios of 84:16 and 65:35, respectively. After incubation for 15 d, no significant difference was observed for NP degradation between composts treated at 45 and 65°C . For both treatments, approximately 80% of NP was degraded at Day 15 and approximately 92% of NP was degraded at Day 43. However, at 25°C , the NP degradation rate was much slower compared with the treatments at the two higher temperatures. By Day 15, between 50 and 60% of NP was degraded for composts treated at

25°C and an additional 55 d was required to degrade $>90\%$ NP. Lower biosolids to wood shaving ratios appear to favor higher NP degradation rates. High temperatures can significantly hasten NP degradation during composting, but composting for longer times at lower temperatures can yield similar results. The data suggest that composting can effectively remove certain PPCPs in biosolids. More studies are needed to monitor the effects of composting on the degradation of other PPCPs in biosolids and to determine the most effective composting treatment parameters.

CONCLUSIONS

The continued growth in human population has created an increase in generation of biosolids, the end product of wastewater treatment plants. The annual production of biosolids in the United States is projected to increase sharply to 8.2 million dry Mg within the next decade (USEPA, 1999). Land application is becoming a major means for biosolids disposal because of its beneficial effects on agricultural productivity of soils. Disposal of biosolids on agricultural fields recycles the nutrients captured from municipal wastewater into agricultural soils, providing valuable nutrients for plants. However, the potential environmental effects of biosolids land application have been a much-debated environmental issue, especially regarding the unknown effects of PPCPs in biosolids.

Biosolids often serve as a sink for PPCPs and their partial metabolites that are not completely degraded during wastewater treatment processes. Research has shown a positive relationship between a compound's hydrophobicity and its sorption on biosolids. Although PPCPs, such as fragrances, flame retardants, surfactants, and their metabolites, have been detected in biosolids, there is limited information on the occurrence of many other PPCPs in biosolids. This lack of information is largely due to analytical limitations because of the complexity of the biosolids matrix. As sensitive analytical

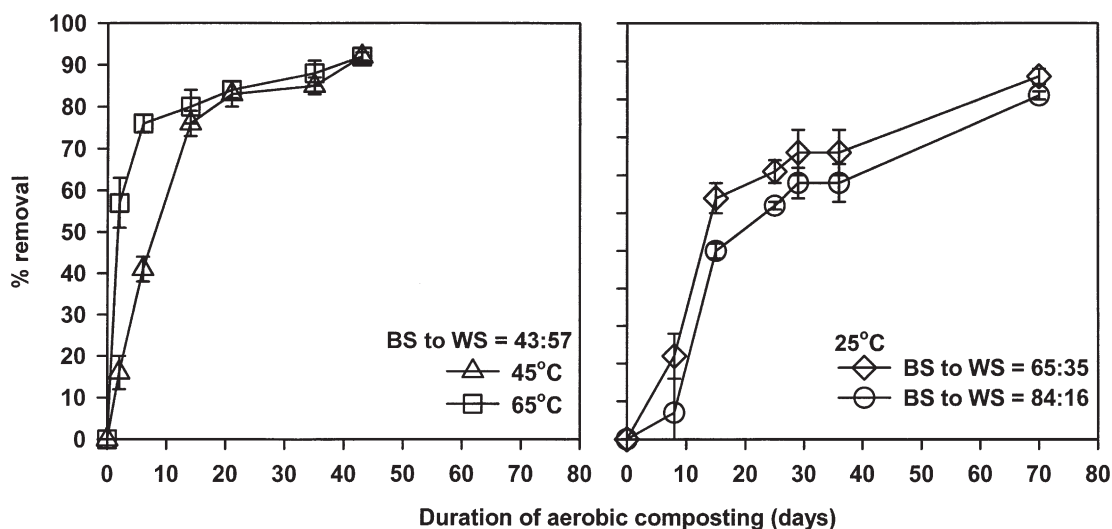


Fig. 6. Removal of nonylphenol (NP) during aerobic composting of biosolids (BS) mixed with wood shavings (WS) at different ratios and different incubation temperatures.

instruments such as liquid chromatography–mass spectrometry and gas chromatography–mass spectrometry become more available and widely used, more PPCPs are expected to be identified and quantified in biosolids.

The few published investigations on the mobility and transport of PPCPs in biosolids-amended soils suggest that land application of biosolids can be a major route through which some PPCPs enter the environment. However, no information is available on how closely the concentrations of PPCPs in the environmental media are related to the land application of PPCP-containing biosolids. Although the characteristics of environmental behavior of PPCPs may be unique because of their close association with biosolids, existing environmental behavior models for many well-studied industrial compounds and agricultural pesticides should be tested on PPCPs of interests. More importantly, the direct effect of land application of PPCP-containing biosolids on organisms in the environment has not been thoroughly investigated. Without detailed information on these issues, assessment of environmental consequence of land application of PPCP-containing biosolids is a difficult task.

To prevent PPCPs from entering the environment, there is an urgent need to document effective wastewater and biosolids treatment techniques. A thorough understanding of transformation mechanisms of PPCPs is the foundation for developing better treatments. Composting can effectively remediate many xenobiotic organic contaminants. Recent pilot-scale composting results suggest that nonylphenol, a metabolite of surfactants commonly used in detergents, can be effectively degraded during biosolids composting. Composting may be effective in removing many other PPCPs in biosolids as well.

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